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Synthetic nitrogen (N) fertilizer has played a key role in enhancing food production and keeping half of the world's population adequately fed. However, decades of N fertilizer overuse in many parts of the world have contributed to soil, water, and air pollution; reducing excessive N losses and emissions is a central environmental challenge in the 21st century. China's participation is essential to global efforts in reducing N-related greenhouse gas (GHG) emissions because China is the largest producer and consumer of fertilizer N. To evaluate the impact of China's use of N fertilizer, we quantify the carbon footprint of China's N fertilizer production and consumption chain using life cycle analysis. For every ton of N fertilizer manufactured and used, 13.5 tons of CO_2-equivalent (eq) (t CO_2-eq) is emitted, compared with 9.7 t CO_2-eq in Europe. Emissions in China tripled from 1980 [131 terrogram (Tg) of CO_2-eq (Tg CO_2-eq)] to 2010 (452 Tg CO_2-eq). N fertilizer-related emissions constitute about 7% of GHG emissions from the entire Chinese economy and exceed soil carbon gain resulting from N fertilizer use by several-fold. We identified potential emission reductions by comparing prevailing technologies and management practices in China with more advanced options worldwide. Mitigation opportunities include improving methane recovery during coal mining, enhancing energy efficiency in fertilizer manufacture, and minimizing N overuse in field-level crop production. We find that use of advanced technologies could cut N fertilizer-related emissions by 20–63%, amounting to 102–357 Tg CO_2-eq annually. Such reduction would decrease China's total GHG emissions by 2–6%, which is significant on a global scale.

carbon accounting | life cycle assessment | food security | policy

The Haber–Bosch process is one of the greatest inventions in modern human history. It enables industrial-scale production of ammonia from atmospheric N_2 using energy. From ammonia, various synthetic nitrogen (N) fertilizers are manufactured, without which nearly half of the world's population would not be alive today (1). However, synthetic N fertilizer has become "too much of a good thing" because much of the N applied to cropland escapes the agricultural system and becomes a pollutant, which disrupts terrestrial and aquatic ecosystem functions and contributes to global climate change. The environmental cost is considerable, between €70 billion and €320 billion per year just for the European Union according to a recent 5-y European nitrogen assessment (2). This 200-member expert panel considered N emission reductions a central environmental challenge in the 21st century and called for a global interconvention N protocol to address the issue. Indeed, coordinated global efforts are particularly critical when dealing with N-related greenhouse gas (GHG) emissions, because such emissions and their impacts recognize no borders.

China is central to the issue. This is not only because China is the largest emitter of fossil-fuel CO_2 into the atmosphere (3) but because China has become a dominating force in the international N fertilizer market. In the past 2 decades (1990–2009), 61% of the world's increase in N fertilizer production and 52% of the increased N consumption occurred in China (4). In 2010, China produced 37.1 terrogram (Tg) of N (Tg N; agricultural consumption of 28.1 Tg N, industrial use of 4.7 Tg N, and export of 4.3 Tg N). This accounted for >30% of world's total and exceeded the combined N fertilizer use in North America (11.1 Tg N) and the European Union (10.9 Tg N) in 2009 (4). Furthermore, China's N fertilizer production and utilization have distinct characteristics. N fertilizer relies heavily on coal as the main source of energy in its production. Coal has a greater carbon footprint than other forms of energy, such as natural gas (Table S1). China's N fertilizer industry is fragmented, consisting of hundreds of small plants with a production capacity only a third to a quarter of typical facilities in developed countries (Table S2). These small enterprises often operate using outdated technologies with relatively low efficiency and high emissions. Perhaps the most striking difference between China and the developed economies is how fertilizer is used in the field. In contrast to the generally mechanized and integrated crop-soil-nutrient management practices widely adopted in developed countries, Chinese farmers hand-apply fertilizer to millions of small plots (Table 1), often resulting in gross overapplication (5). We believe that any global effort in N management must include strong participation by China, and quantifying the carbon footprint of China's N fertilizer chain requires the consideration of conditions specific to China.

Here, we quantitatively evaluate GHG emissions for China's N fertilizer chain through a life cycle analysis beginning from fossil fuel mining as the industry's energy source to postapplication of fertilizers in the field. To do these analyses, we used survey data of 230 fertilizer plants (Table S2) and synthesized literature data of 853 field measurements (Table S3), from which emission factors were derived. We then calculated annual GHG emissions from 1980 to 2010 using statistical data from the China Nitrogen Fertilizer Industry Association (Fig. S1) and estimated future emissions in 2020 and 2030 assuming a 1% annual increment (the same as in the past decade) in N fertilizer demand. Next, we explore emission reduction potential by identifying efficiency gaps between current technologies used in China and more advanced technologies available and by adjusting future N demand based on principles of rational N use that have been proven effective in developed countries and in China. We also discuss socioeconomic factors and propose policy changes that can help curb N-related GHG emissions and assist in moving toward low-carbon agriculture.

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Table 1. Survey results of farmers' practices regarding N fertilizer use in China

Items	Unit	Rice	Wheat	Maize	Fruits	Vegetables
No. of farmers interviewed		4,218	4,554	4,522	6,863	3,889
Synthetic N application	kg N ha^{-1}	209 ± 140*	197 ± 134*	231 ± 142*	550 ± 381*	383 ± 263*
	N applied as urea, %	51%	51%	50%	31%	31%
	N used as a single application, %	9%	26%	13%	16%	22%
	N used before planting, %	50%	60%	49%	—	11%
	N used by hand-broadcasting, %	96%	88%	36%	21%	8%
Manure N	kg N ha^{-1}	15 ± 48*	15 ± 55*	18 ± 52*	42 ± 99*	56 ± 145*
Crop yield	t ha^{-1}	7.2 ± 1.8*	4.9 ± 2.0*	7.4 ± 2.7*	36.7 ± 19.7*	36.0 ± 36.1*
Aboveground uptake	kg N ha^{-1}	122	123	162	128	83
Balance†	kg N ha^{-1}	102	89	87	464	356

This table comprises data taken from responses to a questionnaire survey conducted in 2009 (details are provided in *SI Text*).
*Number following a ± symbol is an SD.
†Balance = Synthetic N + Manure N − Aboveground Uptake.

Results and Discussion

Emission Factors Along the N Fertilizer Chain. For every ton of N produced and used on cropland in China, an average of 13.5 t of CO_2-equivalent (eq) (t CO_2-eq) is emitted (Fig. 1). The largest emission along the chain comes from ammonia synthesis (weighted average of 5.1 t CO_2-eq, 37.8% of 13.5 t). This is partly due to the energy-intensive nature of the chemical engineering process that requires high temperature and pressure and partly due to the low energy efficiency of coal as the main energy source. Coal-based facilities have an emission factor of >5 t CO_2-eq t NH_3-N^{-1} compared with <3 t CO_2-eq t NH_3-N^{-1} for natural gas-based plants (Table S4). For the same energy source, large-scale facilities emit slightly less GHGs per unit of N than medium- or small-scale facilities (Table S4). The next phase involves converting ammonia into various N fertilizer products; the processes have a weighted emission factor of 0.9 t CO_2-eq t N^{-1} but a wide range from 0.3 to 5.7 t CO_2-eq t N^{-1} (Table S5). Thereafter, transport and distribution of the N products have an emission factor averaging 0.1 t CO_2-eq t N^{-1}.

Coal supplies 86% of the energy consumed in the above processes. Methane emissions associated with coal mining have a global warming effect of 11.4 g CO_2-eq MJ^{-1} (10^6 J), compared with <2 g CO_2-eq MJ^{-1} with natural gas or oil (Table S1). We calculated a weighted emission factor of 2.2 t CO_2-eq t N^{-1} for the mining and transport of fossil fuel used in the N fertilizer industry (including 1.8 t CO_2-eq t N^{-1} from mining of the energy used for ammonia synthesis and 0.4 t CO_2-eq t N^{-1} for that used in N product manufacturing). This is 16% of the overall emissions of 13.5 t CO_2-eq t N^{-1}. Neglecting this component would lead to substantial underestimation of China's N fertilizer carbon footprint.

At the end of the chain are GHG emissions from agricultural fields receiving N fertilizers. Weighted for the quantities of N fertilizer used on upland crops and paddy rice systems, the emission factor is 5.2 t CO_2-eq t N^{-1}, including direct emission of N_2O (4.3 t CO_2-eq t N^{-1}) from nitrification and denitrification in soil and indirect emissions (0.9 t CO_2-eq t N^{-1}) calculated from N_2O emission via N deposition (associated with ammonia volatilization), nitrate leaching, and runoff. Our direct emissions are slightly greater, but indirect emissions are substantially less than Europe-based estimates (Table S3). In China, the dominant use of ammonium-based products, together with excessive N application, leads to substantial direct emissions of N_2O (5). As for indirect emissions, China's ammonia loss exceeds that in Europe because of surface spreading and overapplication of ammonia-based N products, but nitrate leaching loss is only a fraction of Europe's (Table S3) because of less nitrate-based products and lower rainfall in most regions of China (6, 7). Our calculations show that upland crop systems emit more GHGs than paddy rice fields, 5.9 t vs. 2.8 t CO_2-eq t N^{-1} (Table S3), which is comparable to Intergovernmental Panel on Climate Change (IPCC) values (6.2 vs. 2.9 t CO_2-eq t N^{-1}).

The overall emission factor we obtained (13.5 t CO_2-eq t N^{-1}) is greater than the estimate for the European N fertilizer chain (weighted average of 9.7 t CO_2-eq t N^{-1}; ref. 7), mainly because of higher emissions associated with coal mining as well as ammonia synthesis and fertilizer manufacture from a general lack of technological advancement in China (discussed elsewhere in this paper). Our results also differ from two previous studies involving China's N fertilizer-chain carbon footprint estimates. One estimated emissions at 9.6 t CO_2-eq t N^{-1} (8), and the other estimated emissions at 15–31 t CO_2-eq t N^{-1} (9). Their numbers were derived from limited data and did not include life cycle analyses.

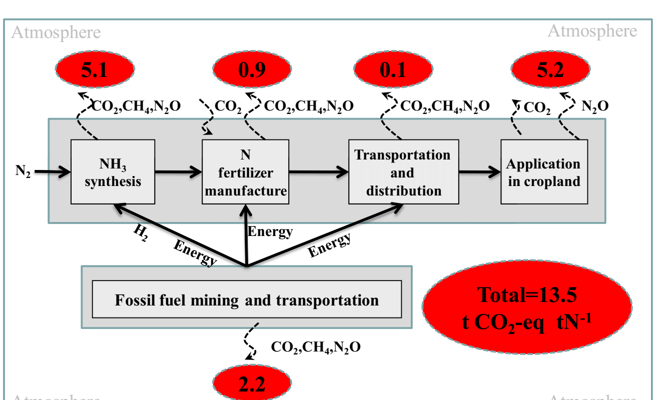

Fig. 1. Life cycle assessment of GHG emissions from manufacturing and field use of N fertilizer in China and weighted emission factors of main processes (system boundaries are described in the main text). Atmospheric nitrogen (N_2) is combined with hydrogen using energy from fossil fuels. The produced NH_3 is reacted with CO_2, nitric acid, hydrochloric acid, or phosphoric acid to produce different N fertilizer products. These fertilizers are transported by various means before being applied to croplands. The solid line represents the materials and N fertilizer flow. The broken line represents GHG exchanges between the fertilizer chain and the atmosphere.

Past, Present, and Future Emissions. Estimated N fertilizer-related GHG emissions in China totaled 131 Tg CO_2-eq in 1980 and increased steadily to 452 Tg CO_2-eq in 2010, with an average increase of 10.7 Tg CO_2-eq·y^{-1} (Fig. 2). This steep increase results directly from N fertilizer production and consumption trends (Fig. S1). In recent years, N-related GHG emissions account for about 7% of total emissions from China (6,100 Tg CO_2-eq in 2004, the most recent data available; ref. 10; Table S6). Assuming a 1% annual increment in agricultural demand for N while maintaining the same export (4.3 Tg N) and industry use (4.7 Tg N) as in 2010, China's N fertilizer demand for agriculture would amount to 33 Tg

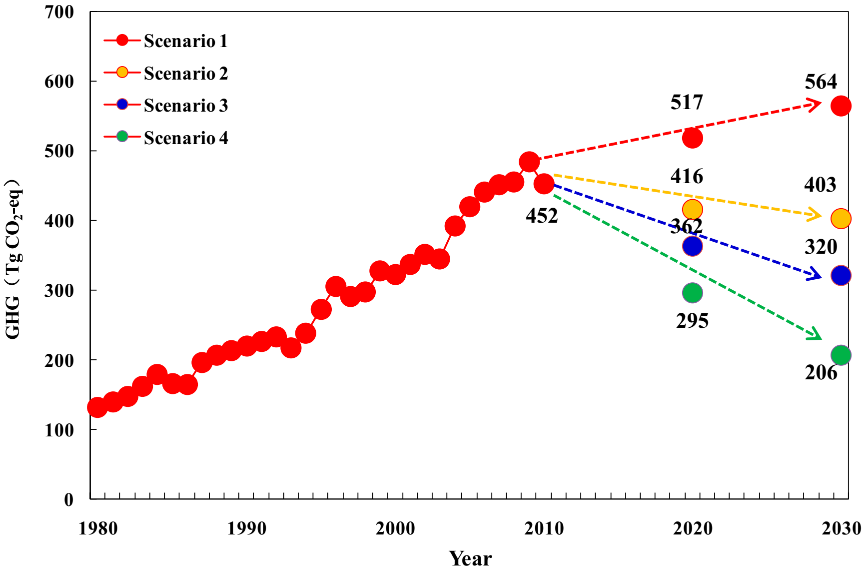

Fig. 2. GHG emissions associated with the N fertilizer chain in China. Emission amounts for 1980–2010 were calculated using emission factors (Fig. 1) derived from a 2005 survey and annual N production and consumption records. Emission estimates for 2020 and 2030 consider four scenarios: scenario 1, business-as-usual; scenario 2, improved manufacturing technologies; scenario 3, improved manufacturing technologies plus controlled N use; and scenario 4, improved manufacturing technologies with reduced N use on croplands.

N in 2020 and 36 Tg N in 2030. Associated GHG emissions would reach 517 Tg CO_2-eq and 564 Tg CO_2-eq, respectively. To put these numbers in perspective, total national GHG emissions from France and Germany in 2009 from all sources were 458 Tg N and 937 Tg CO_2-eq, respectively (11).

N fertilizer has played an indispensable role in doubling crop yields in China during the past 3 decades (12) and is estimated to have contributed to a net gain in soil organic carbon of 85 Tg per year (13). Nevertheless, our data show that N fertilizer-related GHG emissions are several times greater in magnitude than soil organic carbon gains. For China to reduce the gap between GHG emissions and soil carbon sequestration and to move toward low GHG emission agriculture, it is necessary to examine the entire N chain to identify potential emission reductions.

Potential Mitigation. Technological innovation can have a large impact on emission reduction, particularly at the beginning of the N fertilizer chain involving coal mining, ammonia synthesis, and N product manufacturing. For each of these sectors, we compare current technologies used in China with more advanced ones and also with the best technologies available worldwide to estimate emission reduction potential (Table 2).

i) Methane emissions from coal mining operations have a large global warming effect, and their recovery is only 15–23% in China (14, 15), compared with 35% with more advanced recovery technologies or 60% with the best system available (16). Adopting one or another of these would lower the emission factor from the current 0.24 t CO_2-eq t^{-1} coal to 0.20 or 0.14. The emission reduction benefit would extend beyond the N fertilizer industry because coal constitutes 70% of the total energy supplies in the entire country (12).

ii) Coal-fired electricity power plants in China have a heat conversion efficiency of 37–38% with the current subcritical engine units. Emerging technologies can increase the efficiency to 41–42% with supercritical units and to 46–48% with ultra-supercritical units (17). Adopting these new engine units would lower the carbon footprint for electricity from the current 1.12 kg CO_2-eq kilowatt-hour (kWh^{-1}) to 1.08 or 1.03 kg CO_2-eq kWh^{-1}. Again, the benefits would be applicable across the whole economy, and not just in the N fertilizer industry.

iii) The process of making NH_3 from atmospheric N_2 is energy-intensive; current technologies in China have an efficiency averaging 51.3 gigajoule (GJ) t NH_3-N^{-1}, compared with 43.7 GJ or 32.8 GJ t NH_3-N^{-1} with more advanced or the best technologies worldwide (18). Adopting the superior technologies would lower the emission factor from 5.1 to 3.2 or 2.4 t CO_2-eq t NH_3-N^{-1}.

iv) Urea is the main product, and its energy consumption could be lowered from 8.9 to 8.0 or 7.0 GJ t N^{-1} using better or the best available technologies. More dramatic impacts on emissions could potentially be achieved with ammonium nitrate (AN) production. China's AN production facilities mostly use 1960s' technologies, which consume 3.5 GJ t^{-1} N compared with 1.6 GJ t^{-1} N or even less with modern technologies (18). Moreover, AN manufacturing involves converting NH_3 into HNO_3, and the conversion process emits N_2O, currently at 8.0 kg N_2O t

Table 2. Energy use and GHG emissions from N fertilizer manufacture

Items		Unit	Currently in China	Advanced technology	Best technology
Coal mining CH_4 recovery		%	20*	35*	60*
Thermal efficiency at coal-fired power plants†		%	37–38	41–42	46–48
Energy use in NH_3 synthesis		GJ t NH_3-N^{-1}	51.3	43.7‡	32.8‡
Energy use in N product manufacturing	Urea	GJ t N^{-1}	8.9	8.0‡	7.0‡
	AN		3.5	1.6‡	0‡
N_2O emission in AN manufacture		kg N_2O t HNO_3^{-1}	8.0	1.9‡	0.5‡

*Coal bed methane recovery is reported to be 15–23% in China (14, 15); we take 20% as the average. Recoveries for advanced and best technologies are from a US Environmental Protection Agency publication (16).
†Data are from a study by Zhou (17).
‡Data are from a report by the International Fertilizer Association (18), with advanced technologies being the world average and the best technologies being those that operate at the highest energy efficiency.

HNO_3^{-1} in China, whereas an N_2O-abatement technology used elsewhere has lowered the emissions to 1.9 N_2O t HNO_3^{-1}, or even 0.5 kg N_2O t HNO_3^{-1} with the best technology (18). At present, AN is a minor product in China's N fertilizer portfolio (Fig. S1), but if the composition of N products is changed from ammonium-based to nitrate-based (see discussion below), adopting more efficient technologies at the manufacturing stage will be essential.

Combining all four components discussed above, we estimate the emission factor for the N fertilizer industry in China can be reduced from the current 8.3 t CO_2-eq t N^{-1} (2.2 for energy mining + 5.1 for ammonium synthesis + 0.9 for N product manufacture + 0.1 for fertilizer distribution; Fig. 1) to 5.8 or 4.7 t CO_2-eq t N^{-1} with more advanced or the best technologies (Table S7). In performing these analyses, we did not include "carbon capture and storage" technologies currently being tested in Europe and America (19) because these are still a long way from commercial use.

At the end of the N fertilizer chain, there is also considerable scope to reduce emissions resulting from application of fertilizers in the field. Adopting science-based fertilizer application practices is critically important (5, 20), as discussed in subsequent paragraphs. Here, we present some technological- and management-related options (Table 3). First, nitrate-based fertilizers are associated with less N_2O emission than urea or ammonium-based fertilizers (21) because most N_2O is generated from the nitrification process, at least in relatively low-rainfall regions, such as the North China Plain (22). This contrasts with many other regions in the world (and probably other regions in China), where denitrification appears to be the dominant process generating N_2O. Nitrate-based fertilizers also generate less ammonia loss than ammonium-based products (23). Therefore, adjusting the current N product makeup (with 97% being ammonium-based) may help reduce overall N_2O emissions in some regions. However, such a product shift must be preceded by upgrading the AN manufacturing technologies as mentioned previously; otherwise, N_2O emissions during HNO_3 production may exceed potential emission reductions downstream in the field. Second, urea is the main N product in China, and its surface-spreading is associated with considerable N loss via ammonia volatilization (5). Adopting subsurface application can greatly decrease ammonia volatilization, and therefore reduce indirect N_2O emissions (24). Still, possible tradeoffs exist. There may be greater N_2O emissions from nitrification and denitrification of subsurface-applied urea (25). The net effect on emissions will need to be evaluated for different regions, cropping systems, and management practices. Lastly, enhanced efficient fertilizers (including products with surface coatings or incorporating inhibitors of nitrification or urease activity) can improve N use efficiency and substantially reduce GHG emissions; a decrease of 77% in N_2O emissions from using nitrification inhibitors (22) or 60% in NH_3 losses from using urease inhibitors (24) has been reported. The downside of these products is the increased cost incurred (26), making them prohibitive for widespread adoption in grain crops unless incentives are introduced through subsidies or other measures as a means of enhancing environmental services.

Gross overapplication of N fertilizers in China has been well-documented, with a nationwide range of 30–60% above agronomically sound and environmentally sensible recommendations (5). The extent of N overuse is further illustrated by a large-scale survey we conducted recently with >13,000 grain producers and >10,000 fruit and vegetable farmers (Table 1). Excessive N use is widespread: Grain crops receive 220–270 kg N ha^{-1} but remove only 120–160 kg N ha^{-1}, fruits and vegetables receive 400–600 kg N ha^{-1} but remove only 83–130 kg N ha^{-1} (Table 1). The current situation is a result of numerous interacting economic, social, psychological, and policy factors, as discussed in a subsequent section of this paper.

At the national level, total N removal in aboveground crop parts amounted to 16.4 Tg in 2005 (27) and 17.2 Tg in 2010, and it will be 19.0 Tg by 2020 and 21.0 Tg by 2030 assuming a 1% annual increment in crop yield. Our recent work based on long-term intensively managed cropping systems in China shows that the optimum N rate for a crop approximates aboveground crop N removal (28). Applying this N balance concept would suggest that N fertilizer use nationally could be reduced by 42% from current use. Interestingly, the suggested 42% reduction is in line with direct experimental evidence that in two major grain-producing regions (the Yangtze Basin and the North China Plain), N rates can be reduced by 30–60% with no yield loss (5). Also, a rough balance between N fertilizer input and crop removal has been the case in general in developed countries (29).

To integrate the mitigation potentials discussed above and to evaluate their impacts on GHG emissions in coming decades, we performed scenario analyses (detailed data are provided in Table S7), and the results are summarized in Fig. 2. Scenario 1 is business-as-usual, maintaining current technologies and practices and assuming a 1% annual increment in domestic N fertilizer use (as in the past decade). GHG emissions would be 517 and 564 Tg CO_2-eq for 2020 and 2030, respectively. Scenario 2 assumes upgrading industrial technologies to the more advanced level by 2020 and to the best level by 2030 while maintaining the 1% annual increment in N demand. This would result in a net reduction of 102 Tg CO_2-eq by 2020 and 161 Tg CO_2-eq by 2030 compared with scenario 1. Scenario 3 includes the same technological advances as in scenario 2 but keeps N fertilizer at the 2010 level (i.e., no further increase; the rationale for this scenario is discussed later). This would further increase the net reduction (from the base scenario 1) to 155 Tg CO_2-eq for 2020 and to 243 Tg CO_2-eq for 2030. Scenario 4 integrates the technological advances in fertilizer manufacture with more rational N application to crops (achieved using the N balance approach discussed earlier), decreasing N fertilizer use by 21% in 2020 and 42% in 2030 (i.e., a two-step approach to reduce excessive N use). The net reduction (from the base scenario) would be 222 CO_2-eq for 2020 and 357 Tg CO_2-eq for 2030, respectively. There is considerable scope to replace some N fertilizer with livestock manure and probably through better integration of biological N fixation into cropping systems. Thus, further emission reductions are possible, but an in-depth analysis is beyond the scope of this paper.

Table 3. N losses following fertilizer application

Items	Field type	Unit	Currently in China	Advanced technology	Best technology
N_2O emission	Upland	kg N_2O-N kg N^{-1}	0.0105	0.01*	0.007†
	Paddy		0.0041	0.003*	0.003*
NH_3 losses	Upland	kg NH_3-N kg N^{-1}	0.129	0.1*	0.02‡
	Paddy		0.179	0.1*	0.1*
NO_3^- losses	Upland	kg NO_3-N kg N^{-1}	0.098	—	0.04§
	Paddy		0.014	—	—

*Data are from a publication by the IPCC (37) (i.e., IPCC default values, which were mostly derived based on developed economies).
†Data are from a study by Bouwman et al. (21), assuming replacement of urea with $Ca(NH_4)(NO_3)_3$.
‡Data are from a report by the International Fertilizer Association (23), assuming replacing urea with $Ca(NH_4)(NO_3)_3$.
§Data are from a study by Li et al. (39), based on experimental results using nitrate inhibitors.

Overall, the magnitude of potential reductions associated with the various scenarios, ranging from 102 to 357 Tg CO_2-eq, represents a 1.7–5.9% reduction in China's total GHG emissions from all sources (2005 value). This is significant nationally and globally because the feasible emission reductions from improvements in the N fertilizer chain in China are similar in magnitude to the total national reduction goals for 2020, from all sources, sought by several countries [e.g., Germany (365 Tg), France (158 Tg), and the United Kingdom (235 Tg)] (30).

General Discussion

Our analysis, using a life cycle assessment approach, demonstrates that it is essential to include the manufacturing component of the N fertilizer chain (even extending to methane emissions from the mining of coal as an energy source for N manufacture) because these parts of the chain constitute 61% of total emissions (Fig. 1) and provide considerable scope for substantial GHG reductions (scenario 2). China's N fertilizer industry consists of ∼500 companies, as opposed to >200 million individual farmers at the "utilization" end of the chain; thus, it should be easier in the short to medium term to achieve changes in the manufacturing processes through technological innovation and government action. Large capital investment is required for this transformation. One possible solution is for the Chinese government to reallocate the large subsidies, roughly US $7.46 billion during 2008–2009 alone (31), provided to the fertilizer industry through tax breaks and energy subsidies, for technological upgrading of fertilizer plants. Another option is through international intervention via mechanisms, such as carbon trade/credits to accelerate technological advancement. More detailed discussion of the issue and a cost–benefit analysis are beyond the scope of this paper, but we hope this analysis stimulates international interests in upgrading the N fertilizer production chain in China.

China has to grow food to feed >20% of the world's population with only 9% of the world's arable land. Consequently, food security remains the top priority above other concerns unlike the case in developed economies, where national-scale food security is not a major concern (32). This is the basis for scenario 3, where we consider maintaining N fertilizer use at the 2010 level without further increases. This means putting an end to the 50-y trend of increased N production and use. This is not to be taken lightly, because to many, decision makers and farmers alike, continuous growth in agricultural output is thought to depend on increasing fertilizer input. Although still undesirable environmentally, this scenario is probably more likely than scenarios with fertilizer use reductions, given China's political and societal modes.

Clearly, minimizing N fertilizer overuse at the end of the chain is vital. This would not only enhance N fertilizer efficiency and lower emissions in fertilized fields but, more importantly, decrease the total amount of N fertilizer demand. The latter means emission reductions involving the entire N fertilizer chain. Various factors contribute to the excessive N use in China. First, fertilizers have been kept at artificially low prices through heavy government subsidies (31), which obscure the financial burden resulting from excessive N use. Second, there is the absence of an effective and functional extension system that can reliably and systematically deliver science-based recommendations and techniques to hundreds of millions of farmers, although such recommendations have been developed for all major crops and cropping systems in China (33). Third, the land is farmed in small parcels, averaging <0.1 ha per household, which hinders the development and adoption of technologies for mechanized fertilizer application with better control and precision. Fourth, rapid economic development in China has led to the phenomenon of "part-time farmers" because many rural people, especially better educated younger people, are moving into nonfarm work, and this is often more important for household incomes than farming. Consequently, classic models of agricultural extension and assumptions of increasing technical understanding by farmers may no longer be applicable. Improving delivery of technical information at the farm level to enhance N fertilizer use efficiency has value but has been demonstrated to be slow in altering farmer behavior. We propose that alteration of policies related to fertilizer production will be more effective in delivering the necessary changes. Current N fertilizer-related policies were devised decades ago, with the aim of increasing N application for enhanced crop production (*SI Text*). These policies now need to be revised to address both food security and sustainability issues. The huge subsidies to maintain low fertilizer cost for farmers should be replaced with programs that promote environmental services without threatening national food security. For example, incentive programs are needed to improve the management and enhance the utilization of large amounts of livestock manure generated in the nation, which, in turn, would allow substantial reduction of chemical fertilizers (27). Also, payments can be made to cover the additional cost of nitrate-based fertilizer and enhanced efficient fertilizers in situations in which there is clear evidence that these will increase N use efficiency and decrease the amount of N needed. Furthermore, financial support to promote the development of a contractor sector for fertilizer application can be beneficial. Such contractors can (*i*) purchase machinery for subsurface urea application, decreasing ammonia losses; (*ii*) apply N at the "right time," overcoming the labor shortage problem; and (*iii*) comprise a professional group to receive technical information on N fertilizer management.

Conclusions

N fertilizer has been and will continue to be indispensable for China's quest to produce sufficient food to meet its growing demands. However, decades of excessive N use have contributed to a variety of environmental problems, including large GHG emissions and serious water pollution. Our life cycle analysis shows the significance of the carbon footprint associated with the N fertilizer chain in China. GHG emissions tripled from 1980 to 2010, with the amount growing from 131 to 452 Tg $CO_2\text{-eq}\cdot y^{-1}$, and, if unabated, to 564 Tg $CO_2\text{-eq}\cdot y^{-1}$ by 2030. China needs a combination of reforms in the fertilizer industry and changes in management practices and technologies at the farm level to minimize excessive N use in the field. Our scenario analysis indicates it is feasible to reduce GHG emissions by 20–43% from a "business as usual" scenario by 2020 if an appropriate range of mitigation measures are introduced covering both N fertilizer manufacture and its agricultural use. The corresponding reduction by 2030 is 29–63%. Such reductions are in the range of 1.7–5.9% of current national total emissions from all sources. A reduction of this magnitude makes a highly significant contribution to national goals of moving toward a low-carbon economy and is highly significant globally. Minimizing N fertilizer overuse will also deliver "multiple wins" [e.g., improved water quality (with benefits for fish production), enhanced air quality (with associated benefits for human health), less acidification of the soil, improved income for farmers, greater spending power in the rural economy].

Materials and Methods

Life Cycle Assessment Approach for N Fertilizer Chain. We used a life cycle assessment approach to estimate GHG emissions due to the main components of the N fertilizer chain in China, primarily using Chinese-specific parameters rather than IPCC tier 1 default values. According to the International Organization for Standardization's International Standard ISO 14042 (34), the life cycle of N fertilizer should be conducted from "cradle to grave." Therefore, we include GHG emissions associated with mining of fossil fuel used for fertilizer production, transport of fossil fuel, fertilizer synthesis, fertilizer transport and distribution, and gaseous emissions (direct and indirect) when fertilizers are applied to farmland (Fig. 1).

GHG Emission from Fossil Fuel Mining. Two published studies have estimated the GHG emission factors (CO_2, CH_4, and N_2O) in Chinese energy production systems (coal, natural gas, oil, and electricity) using a life cycle assessment approach (14, 35). We used these China-specific emission factors in our study (details are provided in Table S1).

GHG Emission from Ammonia Synthesis. Ammonia is the primary material from which various N fertilizer products are produced. Ammonia synthesis is a major contributor to GHG emissions because of the large energy requirement for its manufacture. The Chinese Nitrogen Fertilizer Industry Association (CNFIA) surveyed 230 companies (Table S2), which account for 40% of the total N fertilizer industry in the nation, including all the large- and medium-scale plants. The survey collected information on the total energy consumption between 2002 and 2005. We have adopted the raw material consumption rate of the ammonia industry determined by this survey and classified the industry into eight categories to estimate different GHG emission factors associated with ammonia synthesis (Table S4).

GHG Emission from N Fertilizer Manufacture. As is the case with NH_3 synthesis, a range of different processes are used in the manufacture of specific fertilizer products. We included five N fertilizer products in this study: urea; AN; ammonium bicarbonate (ABC); ammonium chloride; and compound fertilizers containing N, phosphorus, and potassium (NPKs). We used the specific energy consumption rate of each product determined by the CNFIA survey and by Fan et al. (36) and estimated a GHG emission factor for each (Table S5). The CO_2 fixed during the production of urea and ABC is emitted later into the atmosphere when the fertilizers are applied in the field; thus, it was not included in the calculations.

GHG Emission from Transporting Energy and N Fertilizer Products. We obtained the average transportation distances by train and truck in China for coal, crude oil, and N fertilizer from the National Bureau of Statistics of China (12). We adopted the IPCC (37) default emission factors for N_2O, CH_4, and CO_2 for energy combustion by internal-combustion engines for vehicle transportation (Table S8). Combining these values, we estimate GHG emission factors for energy and fertilizer transportation (details are provided in *SI Text*).

GHG Emission from Postapplication Field. The GHG emissions caused by N fertilizer applied to croplands are mainly in the form of N_2O, including direct and indirect emissions. We classified Chinese agricultural land into two groups: upland fields and paddy fields. We compiled all published field measurements in China (a total of 853) and summarized the results using a meta-analysis method to derive direct and indirect N_2O emission factors. Direct emission factors for upland fields and paddy fields were obtained from a study by Gao et al. (38), which includes 456 N_2O emission measurements in China (195 paddy fields and 261 upland fields). Indirect emissions include N_2O resulting from N deposition (associated with NH_3 volatilization) and NO_3^- leaching. We summarized 397 published field measurements (138 paddy fields and 259 upland fields) from 47 literature sources. We used IPCC (37) values for the proportion of those losses emitted as N_2O (Table S8). Then, we calculated the GHG emission factors for paddy fields and upland fields, respectively (Table S3).

Total GHG Emissions from N Fertilizer Production and Utilization. We calculated annual total GHG emissions from N fertilizer production and consumption in China from 1980 to 2010. The emission factors for the various sectors (energy mining and transport, NH_3 synthesis, fertilizer manufacture, N products distribution, and N application) were multiplied by the respective quantities of the materials to derive the amounts of sector-specific emissions, which were then summed for each year (details are provided in *SI Text*).

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Supporting Information

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SI Text

Background of Nitrogen Fertilizer in China. ***Survey of farmers' practices regarding nitrogen fertilizer use.*** The fragmented and small-scale farming and diverse cropping systems in China make it difficult to describe fully the real practices regarding nitrogen (N) fertilizer use by >200 million farmers. However, this information becomes increasingly important to enhance N management for agricultural development and for environmental protection. In 2001, 2003, 2008, and 2009, China Agricultural University organized four farmer surveys on a national scale in cooperation with extension service departments and agricultural universities in 27 provinces (totaling 32 provinces plus Hong Kong, Macao, and Taiwan in China). Each survey solicited and trained interviewers, used a uniform questionnaire, and targeted 12 major crops in the country (wheat, maize, rice, cotton, soy, oil crops, sugar crops, tobacco, tea, potato, vegetables, and fruits). These crops represent more than 95% of the total planting area. For each crop, the major production regions were required to select three to five counties per province, three towns per county, and two villages per town according to regional farmers' average income. In each village, 8–10 households were randomly selected and interviewed. In this way, we try to ensure the results are representative of the whole of China.

The results of the 2001, 2003, and 2008 surveys have been published (1–3). The results from the 2009 survey were used in the current study, including 2,346 villages in 27 provinces of China. Here, we concentrate on five major crops (Table 1) because these crops accounted for 82% of N fertilizer consumption in China in 2009. For each crop, we calculated a national weighted average value for N fertilizer use according to the planting area of each sample. It needs to be mentioned that fruit farmers reported growing 1 or more of 11 types of fruits/orchards (e.g., peaches, pears, apples, grapes, oranges, tangelos, bananas, chestnuts) and vegetable growers reported growing 1 or more of 17 types of vegetables (e.g., cucumbers, tomatoes, cabbage, eggplant, beans, potatoes, celery, garlic).

The results show that Chinese farmers applied excessive amounts of N fertilizer compared with the aboveground uptake. The latter includes "removal in harvested product" and "removal in aboveground residues (e.g., leaves, stems, etc.)." This was calculated based on yields and unit of N uptake (17, 25, 22, 3.5, and 2.3 kg of N per 100 kg of aboveground mass of rice, wheat, maize, fruits, and vegetables). The unit of N uptake values was taken from weighted averages of regional cropping systems from the work of Zhang et al. (4). For grain crops (wheat, maize, and rice), the average value of the rate of N fertilizer application in this study is in line with other research, for example, 237 ± 70 kg N per hectare for maize in a study by Chen et al. (5). However, farmer practice varied greatly; 86% farmers had a rate of N fertilizer application far exceeding the aboveground uptake, whereas 14% farmers applied less N fertilizer than the aboveground uptake. For vegetables and fruits, the rate of N application of all the farmers exceeded the aboveground uptake.

Survey results also indicate the problem of wrong application time [e.g., for wheat, 60% of the N was applied before planting and 26% of farmers reported single application times (i.e., no split applications)]. Also, hand-broadcasting (surface-spreading fertilizers before soil preparation or irrigation) is still a dominant means of fertilizer application, especially for wheat. For maize, hand-applying fertilizer into small holes drilled near the plants is a common practice. For vegetable and fruits, fertilization with flood irrigation or band application (spreading fertilizer along a ditch) is common.

Statistics of N fertilizer in China. From 1980 to 2010, both the manufacture and use of N fertilizer in China increased steadily (Fig. S1). In 2010, total ammonia production was 40.9 terrogram (Tg) of NH_3-N, 3.3-fold greater than in 1980. The total production of N fertilizer in 2010 was 37.1 Tg N, 3.7-fold greater than in 1980. The proportion of N fertilizer products also changed substantially during this period, with urea increasing from 30% to 68%, whereas ammonium bicarbonate (ABC) decreased from 51% to 11%. The production of other fertilizer products (mainly compound fertilizers) also increased rapidly. In terms of consumption, 28.1 Tg N fertilizer was used in agriculture in 2010 (arable land, forestry, and aquatic systems), nearly 3.2-fold that in 1980. Another notable change in the past 3 decades is the situation regarding imports and exports of N fertilizer. N imports decreased from 1.43 Tg in 1980 to 0.3 Tg in 2010, but N exports increased from 0 in 1980 to 4.6 Tg in 2010. Associated emissions generated in the manufacture of exported N were included in our calculations, although emissions associated with use of the exported N outside of China were not.

N fertilizer-related policies in China. The development of China's fertilizer industry and field application is entwined with and affected by governmental policies, which have evolved over the past 6 decades and can be categorized chronologically into the following: a historically strict state ownership and price control system (1949–1984, phase I) gradually transitioned into a dual system of state control plus market adjustment (1985–1997, phase II), followed by the government releasing its direct grip and letting the market influence fertilizer production and prices while still maintaining indirect intervention through a price cap (1998–2009, phase III), and then moving to a full market system with the price cap removed (except for imported fertilizers; 2009, phase IV).

Although the government has removed the price cap, various subsidy programs on fertilizer production and consumption remain intact. Five major subsidy programs are massive in scale and most influential, including the following:

i) Subsidies on electricity use at fertilizer manufacturing plants, which were implemented through preferential pricing granted by the National Development and Reform Commission (NDRC). In 2003 and 2006, the NDRC instituted electricity price hikes for all industrial and commercial entities except for the fertilizer industry.

ii) Subsidies on fertilizer transport, through exemptions from the railway construction fund [a charge of US \$0.0005 per ton (t)$\cdot$km^{-1} instituted by the Ministry of Railways in 1995 and still in effect today] and charging a lower price for fertilizer transportation than for that of other commodities.

iii) Exemption from value added tax (VAT), which constituted by far the largest subsidy program for the fertilizer industry. Before 2004, a reimbursement program was carried out by returning 50% of the VAT back to fertilizer enterprises. During 2004 and 2005, a test was run to exempt the VAT entirely for urea. Then, starting in 2006, VAT exemption was granted for all fertilizer products.

iv) There was also a fertilizer reserve subsidy program put in place by the NDRC, aiming at stabilizing the fertilizer supply and minimizing a peak season shortage.

v) In addition to the above four subsidies for fertilizer producers, the government has subsidized farmers since 2006.

This subsidy was designed to eliminate the negative effect of the changing fertilizer price on grain production. It was estimated that China's fertilizer subsidies totaled US $7.46 billion in 2008/2009 alone (6).

SI Materials and Methods

Greenhouse Gas Emission from Fossil Fuel Mining. Intergovernmental Panel on Climate Change (IPCC) guidelines (7) provide default factors for greenhouse gas (GHG) emissions from energy mining, representing a global average, but they may not be suitable for specific situations of coal mining in China. Yuan et al. (8) estimated GHG emissions during primary energy production in China using a life cycle method. Their study monitored large energy plants and took into consideration the energy consumption of boilers and vehicles, as well as CH_4 recovery in coal mines. They estimated average emission factors associated with mining processes to be 0.24 t of CO_2-equivalent (eq) t^{-1} coal, 0.07 kg CO_2-eq m^{-3} natural gas, and 0.08 t CO_2-eq t^{-1} oil. Ma (9) evaluated GHG emissions from electricity generation using coal, hydro, and nuclear power in China and obtained a weighted average GHG emission factor for electricity generation of 1.12 kg CO_2-eq $kW{\cdot}h^{-1}$. We used these China-specific emission factors in the present study (Table S1).

Different fossil fuel or energy forms have different calorific values (e.g., 20,908 $kJ{\cdot}kg^{-1}$ for coal, 38,931 $kJ{\cdot}m^{-3}$ for natural gas, and 3,596 $kJ{\cdot}kW{\cdot}h^{-1}$ for electricity; Table S8). We calculated the carbon footprints per unit of calorific value to be 11.4, 1.9, 1.9, and 311.8 g CO_2-eq MJ^{-1} for coal, natural gas, crude oil, and electricity, respectively (Table S1).

GHG Emission from Ammonia Synthesis. According to the statistical data of the Chinese Nitrogen Fertilizer Industry Association (CNFIA), there were 570 ammonia companies nationwide in 2005 (the latest year for which published data are available) with a total production capacity of 46.3 Tg NH_3. For the same year, fuel use at these plants was 70% coal and 22% natural gas, and the remaining fuel used was crude oil. The company size and its annual ammonia production vary greatly, as do the process technologies and raw materials used. Consequently, energy consumption and GHG emissions vary considerably among different plants. In 2005, the CNFIA conducted a survey of 230 companies (Table S2), 40% of the total, including all the large- and medium-scale plants. The survey collected information on the consumption of fossil fuel energy, electricity use, and total energy consumption between 2002 and 2005. We adopted the raw material consumption rate of the ammonia industry based on survey results and classified the industry into eight categories to estimate the different GHG emission factors associated with ammonia synthesis (Table S4). The emission factor for each type of ammonia plant was calculated using the amount of energy consumption from the survey and IPCC default values (Eq. **S1**):

$$GHG_{Ami} = \sum W_{Mij} \times (R_{CO2} + R_{N2O} \times 298 + R_{CH4} \times 25)_{ij}. \quad \textbf{[S1]}$$

GHG_{Ami} is the GHG emission factor of ammonia production (t CO_2-eq t N^{-1}) in the *i*th type plant. The different types of factories are defined in the footnotes of Table S4. W_{Mij} is the consumption of raw material per unit of ammonia N produced (t·t N^{-1}) (j_1 = coal, j_2 = oil, j_3 = gas, and j_4 = electricity). R_{CO2}, R_{N2O}, and R_{CH4} are the CO_2, N_2O, and CH_4 emission factors, respectively, of raw material combustion (t·t N^{-1}), using default values from the IPCC (7). The global warming potentials used for converting CH_4 and N_2O to CO_2 equivalents were 25 and 298, respectively (ref. 10; Table S8).

GHG Emission from N Fertilizer Product Manufacturing. We used data from the CNFIA survey of fertilizer plants to provide information on the different technologies and quantities of each type of fertilizer produced: urea, ammonium nitrate (AN), ABC, and ammonium chloride (ACL). Compound fertilizers with N, phosphorus, and potassium in different formulations (NPKs), which are made using ammonia or other forms of N fertilizer, are gaining in popularity in China. The industry survey lacked data on this, so we used energy consumption data for four major types of compound fertilizer from the *Chinese Fertilizer Manual* compiled by the Chinese Chemical Industry Information Centre (11). We summarized these data and recalculated the GHG emission factor for each N fertilizer product according to Eq. **S2**:

$$GHG_{PNik} = \sum W_{Mijk} \times (R_{CO2} + R_{N2O} \times 298 + R_{CH4} \times 25)_{ijk} + W_{ikN2O} \times 298. \quad \textbf{[S2]}$$

GHG_{PNik} is the GHG emission factor of each product (t CO_2-eq t N^{-1}) (k_1 = urea, k_2 = AN, k_3 = ABC, k_4 = ACL, and k_5 = NPK). W_{ikN2O} is the N_2O emission from producing nitric acid (t·t N^{-1}; nitric acid is one of the raw materials of AN). When NH_3 is converted to urea and ABC, some CO_2 is fixed into the N products. We used the values 1.5 t CO_2 fixed t urea-N^{-1} and 3.6 t CO_2 fixed t ABC-N^{-1} taken from the *Chinese Fertilizer Manual* (11). The CO_2 fixed during the production of these fertilizers is released into the atmosphere after field application, thus cancelling out the earlier fixation (Table S5).

GHG Emissions from Transporting Energy Materials and N Products. Energy consumption by railway locomotives is 2.46 kg of diesel oil per 1,000 t of freight per kilometer for an internal combustion engine (45% of journeys) and 11.0 kW·h per 1,000 t of freight per kilometer for an electric-powered locomotive (55% of journeys) (12). We adopted the IPCC (7) default emission factors for N_2O, CH_4, and CO_2 for energy combustion by internal combustion engines for vehicle transportation. Combining these values, we estimate GHG emission factors of 11 t CO_2-eq per 1 million t^{-1} of freight per kilometer by train and 179 t CO_2-eq per 1 million t^{-1} of freight per kilometer by truck. The average transportation distances by train in China for coal, crude oil, and N fertilizer are 607 km, 912 km, and 1,365 km, respectively (12). The additional transportation distances by truck are not available, but the mean distance of 69 km for all commodities was assumed (12).

Based on the average transportation distance of fertilizer by train and truck, we derived the GHG emission factor 0.027 t CO_2 per ton of products for fertilizer transportation. Different fertilizer products have different N contents (Table S8). Therefore, we estimated the GHG emission factor per ton of N for each fertilizer, such as 0.06 t CO_2 t urea-N^{-1}, 0.07 t CO_2 t AN-N^{-1}, 0.15 t CO_2 t ABC-N^{-1}, 0.11 t CO_2 t ACL-N^{-1}, and 0.18 t CO_2 t NPK-N^{-1}. Weighted by the proportion of different products in total N production (Fig. S1), we obtained 0.1 t CO_2 per ton of fertilizer N emissions in transportation in China. To calculate emissions for transporting fossil fuel, we included coal and crude oil but assumed the emission to be 0 for natural gas (pipe transport), electricity (power line transport), and steam (on-site production and consumption). Accordingly, we determined energy transport emission factors to be 0.019 t CO_2-eq per ton of coal and 0.022 t CO_2-eq per ton of crude oil (Table S1).

GHG Emission from N Fertilizer Application to Croplands. The GHG emissions resulting from N fertilizer applied to croplands are mainly in the form of N_2O, including direct and indirect emissions. China-specific direct emission factors for N_2O from croplands used in this research were taken from Gao et al. (13), who summarized recent field emission measurements from 195 paddy fields (rice) and 261 upland fields (including wheat, maize, soybean, cotton, and peanut; Table S3). The resulting N_2O emissions (0.41% for paddy fields and 1.05% for upland fields) are slightly higher than IPCC default values (0.3% for paddy fields and 1% for upland fields;

ref. 7). Rates of ammonia volatilization and N leaching plus runoff were derived from nonlinear regression equations based on 138 experimental treatments in paddy fields (rice) and 259 experimental treatments in upland fields (including wheat, maize, and vegetables) from an array of publications (14–60). There were 26 data points from paddy fields and 45 data points from upland fields for NH_3 volatilization and 112 and 214 data points from paddy fields and upland fields, respectively, for nitrate leaching. For each cropping system (spring maize, summer maize, winter wheat in the north and south regions, rice in the north and south regions, and vegetables grown in greenhouses and in open fields), a nonlinear regression model describing the emission factor and N application rate was established for individual cropping systems. N fertilizer application rates were calculated using the national average rates for each crop system. Then, a weighted mean for paddy fields and upland fields was obtained taking into account the acreage for individual cropping systems. The N leaching rate from fertilizers applied in this study is much lower than IPCC default values (30% of total N) because leaching is generally low in China due to low rainfall in many regions. The ammonia volatilization rate applied in this study is slightly higher than the IPCC default value (10% of total N) because of high losses by this mechanism resulting from surface application of urea and ABC.

Total GHG Emissions from the N Fertilizer Chain. To calculate the total GHG emission from N fertilizer production and consumption in China, we used the annual ammonia synthesis and N fertilizer production data from the CNFIA. In 2010, total ammonia production was 40.9 Tg N. Total N fertilizer production was 37.1 Tg N (including urea, AN, ABC, ACL, NPK produced from NH_3, and other products), of which 4.6 Tg N was exported. These data are very similar to the statistical data of the International Fertilizer Association ([61), but with more detailed information for each product. Therefore, we adopted these data. Relevant emissions from the manufacture of exported N are included as emissions from China, although emissions from its subsequent use outside of China are excluded.

A range of different estimates have been published for fertilizer consumption in China: The United Nations Food and Agriculture Organization (62) cited an estimate of 36.9 Tg N, and the International Fertilizer Association (61) cited an estimate of 33.6 Tg N for 2009, whereas the China Statistics Bureau (12) reported that 23.3 Tg N was consumed in the form of synthetic N fertilizers and 17.0 Tg total nutrients was consumed in the form of compound fertilizers in 2009. It is difficult to separate the composition of compound fertilizers into distinct quantities of N, phosphorus (P), and potassium (K). Also, these data sources usually include N used on arable land but not that used in forestry and aquatic systems. To include N used in these systems in addition to arable land, we calculated the apparent N consumption (production + import − export) using fertilizer production data from the Chinese Nitrogen Fertilizer Industry Association and N trade data from the General Administration of Customs. This gave an estimate of total fertilizer consumption in China of 32.8 Tg N for 2010. The CNFIA estimated that 4.7 Tg N was used in industry; thus, this was subtracted, resulting in a value of 28.1 Tg N used in all agricultural systems (arable land, horticulture, forestry, and aquatic systems). Zhang et al. (1) and Li et al. (2) estimated the distribution of N in the above systems according to large-scale farmer surveys ($n >$ 30,000) in 2001, 2003, and 2008. Using this information, we estimated that the N used in paddy fields was 22% of the total (paddy rice plus aquatic systems) and that used in other upland fields was 78% of the total.

Using this information, we calculated the total annual GHG emission associated with N fertilizer in China from 1980 to 2010 (Eq. **S3**):

$$\begin{aligned}GHG_{total} = {} & AP \times EF_{Ami} + \sum PP_{ik} \times EF_{PNik} + \sum PP_{ik} \times EFt \\ & + (EFdp + EFi) \times PCp + (EFdu + EFi) \\ & \times PCu + (AP \times ENF1 + PP_{ik} \times ENF2_{ik}) \times EFm,\end{aligned} \quad \textbf{[S3]}$$

where GHG_{total} is total GHG emission from N fertilizer production and use in China in specific years, AP and PP_{ik} are ammonia production and each type of N fertilizer production in specific years; PCp and PCu are N consumption in paddy fields (including aquatic systems) and upland fields; EF_{Ami}, EF_{PNik}, and EFt are emission factors in ammonia synthesis plants, fertilizer manufacturing plants (Eqs. **S1** and **S2**), and fertilizer transportation; EFdp and EFdu are direct emission factors in paddy fields and other croplands; EFi is the indirect emission factor (from ammonia volatilization and leaching); ENF1 and $ENF2_{ik}$ are energy consumption rates in ammonia production and different types of N fertilizer production; and EFm is the emission factor in different types of energy mining and transportation. The results for N-related GHG emissions from 1980 to 2010 are presented in Fig. 1.

The Chinese government reported national total fossil fuel consumption and GHG emission for 2004 base (ref. 63; details are provided in Table S6). To estimate the contribution of N-related emissions to the national total emission, we used detailed information on energy consumption and relevant GHG emissions along the N fertilizer chain for 2005 base (Table S6). Results show that total fossil fuel energy consumption associated with ammonia synthesis, fertilizer manufacturing, and fertilizer transportation was 2.6 million gigajoule (GJ) in 2005, accounting for 3.7% of the national total. During mining and final use of this energy for fertilizer production and transportation, 231.7 Tg CO_2 and 805.9 gigagram (Gg) CH_4 were emitted, which constitute 4.6% and 2.8% of the national total emission, respectively. From fertilized fields, 93.6 Gg N_2O was emitted through direct and indirect processes, accounting for 50.0% of the national total. Finally, the total emission was calculated to be 416.8 Tg CO_2-eq after converting CH_4 and N_2O into CO_2-eq, which is 6.8% of the national total.

We chose the years 2020 and 2030 for scenario analysis for several reasons. The international community generally has GHG emission reduction targets for 2020 (64). Similarly, China's national plan for science and technology development is for the period of 2006–2020 (65). However, China's population is expected to peak in the mid-2030s (66), and the demand for food will continue to grow at least until the 2030s. Producing more food and simultaneously decreasing fertilizer use is a huge challenge (67). We presume a reasonable two-tier approach with a full-scale reduction in excessive N use by 2030 but a halfway reduction by 2020.

Scenario 1 assumes "business as usual" with a 1% annual increment in N fertilizer demand. This is perhaps the most probable case unless the country takes serious actions to curb N fertilizer-related GHG emissions. Scenario 2 considers the adoption of more advanced technologies at the manufacturing stage by 2020 and the best stage by 2030 while maintaining the current N use pattern with a 1% annual increment. Intuitively, technologically upgrading the nation's 500 or so fertilizer plants would be more easily achieved than changing the fertilizer use habits of 200 million farmers. This option is unlikely to encounter much resistance among decision makers, who are generally cautious about any measures that might cause food security concerns. Scenario 3 assumes technological advancement while maintaining N fertilizer use at 2010 levels. We consider this option viable, given the widely recognized fact of N fertilizer overuse in China and well-established scientific principles for moving to a more rational use of N fertilizer. Scenario 4 represents the most desirable situation, in which technological advancement is integrated with enhanced field use efficiency while reducing excessive N use. This would require an all-out effort in the country. Datasets used for the various scenarios are presented in Table S7.

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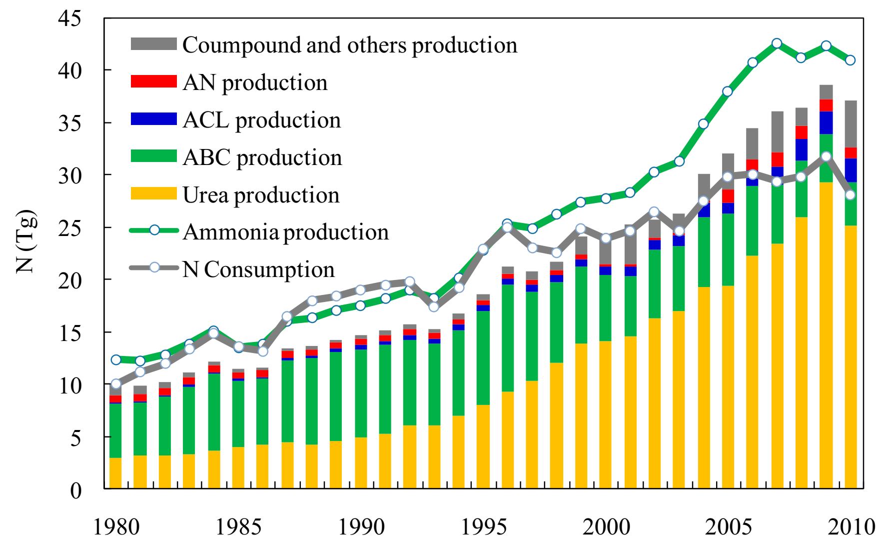

Fig. S1. Changes in N fertilizer production and agricultural consumption from 1980 to 2010 in China. Data from the China Nitrogen Fertilizer Association are cited.

Table S1. GHG emissions during energy mining and transportation

	GHG emission during energy mining		GHG emission during energy transportation	
Energy type	Unit of energy	g CO_2-eq MJ^{-1}	Unit of energy	g CO_2-eq MJ^{-1}
Coal*	0.24 t CO_2-eq t^{-1}	11.4	0.019 t CO_2-eq t^{-1}	0.91
Natural gas*	0.07 kg CO_2-eq m^{-3}	1.9	—	—
Oil*	0.08 t CO_2-eq t^{-1}	1.9	0.022 t CO_2-eq t^{-1}	0.53
Electricity[†]	1.12 kg CO_2-eq kW·h^{-1}	311.8	—	—
Steam[‡]	0.026 t CO_2-eq t^{-1}	9.7	—	—

*Recalculated data are from a study by Yuan et al. (8).
[†]Recalculated data are from a doctoral dissertation by Ma (9).
[‡]Assuming steam produced from coal and recalculated based on emissions from coal and lower heating values of steam.

Table S2. Statistical information on sizes and types of N fertilizer factories in China in 2005

Factory types	Size	Energy source	Total no. of plants	Total production capacity, Tg NH_3	Number of plants in survey: Ammonia synthesis	N fertilizer manufactured: Urea	AN	ABC	ACL
C_1	Large	Coal	2	0.58	2	1	1	0	0
C_2	Large	Gas	18	6.40	18	18	3	0	0
C_3	Large	Oil	7	1.43	7	6	1	0	1
C_4	Medium	Coal	37	5.14	37	27	11	5	0
C_5	Medium	Gas	11	1.59	11	9	5	0	0
C_6	Medium	Oil	3	0.59	3	0	0	0	0
C_7	Small	Coal	443	29.44	141	70	0	101	7
C_8	Small	Gas	49	1.13	11	4	0	7	1
Total			570	46.30	230	135	21	113	9

Large-scale, medium-scale, and small-scale companies are defined by annual NH_3 production of ≤80 kilotonne (kt), >80 kt to ≤180 kt, and >180 kt, respectively. C_1, large-scale corporation using coal as raw material; C_2, large-scale corporation using gas as raw material, C_3, large-scale corporation using oil as raw material; C_4, medium-scale corporation using coal as raw material; C_5, medium-scale corporation using gas as raw material; C_6, medium-scale corporation using oil as raw material; C_7, small-scale corporation using coal as raw material; C_8, small-scale corporation using gas as raw material.

Table S3. GHG emission factors for N fertilizer applied to croplands in China

Data sources	Crop systems	Direct N_2O*: No. of measurements	N_2O, %	EF(t CO_2-eq t N^{-1})	Indirect N_2O†: No. of measurements	NH_3 volatilization, %	Leaching and runoff, %	EF(t CO_2-eq t N^{-1})
China	Paddy field	195	0.41	1.92	138	17.9	1.4	0.89
	Upland	261	1.05	4.92	259	12.9	9.8	0.95
	Weighted average‡			4.26				0.93
IPCC default§	Paddy field		0.30	1.41		10.0	30.0	1.52
	Upland		1.00	4.68		10.0	30.0	1.52

*Direct N_2O emission data in China were cited from a study by Gao et al. (13).
†Indirect N_2O includes N_2O from ammonia volatilization and nitrate leaching and runoff. Data in China were summarized from refs. 14–60.
‡Weighted by the proportion of N fertilizer used in paddy fields (22%) and dry land (78%).
§IPCC default direct and indirect N_2O emission data are from an IPCC publication (7).

Table S4. Energy efficiency and GHG emissions for ammonia synthesis in different types of plants in China

Factory types	Energy efficiency (GJ t NH_3-N^{-1})	GHG emissions (t CO_2-eq t NH_3-N^{-1})*: During combustion in ammonia plants	During energy mining and transportation	Total
C_1	53.5	6.6	0.8	7.4
C_2	44.7	2.4	0.1	2.5
C_3	50.6	3.6	0.4	4.0
C_4	60.9	6.8	2.5	9.5
C_5	52.7	2.8	1.1	3.9
C_6	60.1	4.2	0.8	5.1
C_7	50.6	5.6	2.3	7.9
C_8	56.1	2.9	1.5	4.4
Weighted average†	51.3	5.1	1.8	6.9

GJ, gigajoule.
*GHG emissions from fossil fuel combustion in ammonia synthesis and GHG emissions from fossil fuel mining and transportation are included in this table.
†All the results were weighted by ammonia production in each plant.

Table S5. Energy efficiency and GHG emissions during conversion of ammonia to N fertilizer in different fertilizer plants in China

N fertilizer products	Energy efficiency ($GJ\ t\ N^{-1}$)	GHG emissions(t CO_2-eq t N^{-1}) During energy combustion in fertilizer plants	N_2O in nitric acid	During energy mining and transportation	Total	CO_2 fixed* (t CO_2-eq t N^{-1})
Urea	8.9	0.8	—	0.4	1.2	1.5
AN	3.5	0.3	5.4	0.2	5.9	—
ABC	0.8	0	—	0.3	0.3	3.6
ACL	1.0	0	—	0.3	0.3	—
NPK†	3.2	0.3	—	0.2	0.5	—
Weighted average	6.7	0.7	—	0.4	1.1	—

GJ, gigajoule.
*CO_2 fixed during production of urea and ABC will be emitted to the atmosphere when these products are applied to croplands; thus, they have not been included in the total emissions calculated.
†Energy consumption includes electricity, steam, and coal. We cited the energy consumption rate from four main kinds of products, as reported by Fan et al. (11), and then recalculated the GHG emissions.

Table S6. Total energy consumption and GHGs from manufacturing and N fertilizer use in China in 2005

	Fossil fuel use, million GJ	GHG emission CO_2, Tg	CH_4, Gg	N_2O, Gg	All in CO_2-eq, Tg CO_2
Energy mining and transportation	0.4	49.8	792.7	6.5	71.6
Ammonia synthesis	1.9	162.1	14.8	2.2	163.1
Fertilizer manufacture	0.2	18.4	1.9	24.6	25.7
Fertilizer transportation	0.0	1.5	0.1	0.2	1.6
Direct emissions after application				426.2	127.0
Indirect emissions after application				93.6	27.9
a: Sum of above (related to N fertilizer)	2.6	231.7	809.5	553.3	416.8
b: National total in 2004*	70.8	5,070	28,800	1,107	6,100
a/b, %	3.7	4.6	2.8	50.0	6.8

GJ, gigajoule.
*Data are from a report by The Central People's Government of the People's Republic of China (63).

Table S7. Data used for scenario analysis for GHG emissions in 2020 and 2030

		Business as usual in 2005	Advanced technologies in 2020	Best situation in 2030
Total N balance, Tg	Fertilizer production	32.0	32.0	30.0
	Fertilizer use in agriculture	27.9	23.0	21.0
	N uptake by crop aboveground part	16.4*	19.0	21.0
	Organic N	9.3*	20.5†	20.5†
Emission factors in N fertilizer chain (t CO_2-eq t N^{-1})‡	Energy mining	2.2	1.9	1.7
	NH_3 synthesis	5.1	3.2	2.4
	Fertilizer production	0.9	0.7	0.6
	Fertilizer transportation	0.1	0.1	0.1
	Direct N_2O from field	4.3	4.0	2.9
	Indirect N_2O from field	0.9	0.8	0.2

*Data are from a study by Ma et al. (1).
†Assuming 80% recycling of current organic N sources, like that in European Union countries (2).
‡Emission factors for 2020 and 2030 are derived from Tables 2 and 3 and Eqs. **S1–S3**.

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Table S8. Parameters used for calculations in this study

Parameters	Value	Unit	Ref.
GWP of CH_4	25	kg CO_2 kg^{-1}	[1]
GWP of N_2O	298	kg CO_2 kg^{-1}	[1]
EF of CO_2 for anthracite burning	98,300	kg·TJ^{-1}	[2]
EF of CH_4 for anthracite burning	10	kg·TJ^{-1}	[2]
EF of N_2O for anthracite burning	1.5	kg·TJ^{-1}	[2]
EF of CO_2 for oil burning	73,300	kg·TJ^{-1}	[2]
EF of CH_4 for oil burning	3	kg·TJ^{-1}	[2]
EF of N_2O for oil burning	0.6	kg·TJ^{-1}	[2]
EF of CO_2 for natural gas burning	56,100	kg·TJ^{-1}	[2]
EF of CH_4 for natural gas burning	1	kg·TJ^{-1}	[2]
EF of N_2O for natural gas burning	0.1	kg·TJ^{-1}	[2]
EF of CO_2 for diesel oil burning (highway)	74,100	kg·TJ^{-1}	[2]
EF of CH_4 for diesel oil burning (highway)	3.9	kg·TJ^{-1}	[2]
EF of N_2O for diesel oil burning (highway)	3.9	kg·TJ^{-1}	[2]
EF of CO_2 for diesel oil burning (railway)	74,100	kg·TJ^{-1}	[2]
EF of CH_4 for diesel oil burning (railway)	4.15	kg·TJ^{-1}	[2]
EF of N_2O for diesel oil burning (railway)	28.6	kg·TJ^{-1}	[2]
Lower heating values of coal	26,344	kJ·kg^{-1}	[3]
Lower heating values of oil	41,816	kJ·kg^{-1}	[3]
Lower heating values of natural gas	38,931	kJ·m^{-3}	[3]
Lower heating values of diesel oil	42,652	kJ·kg^{-1}	[3]
Lower heating values of electricity	3,596	kJ·kW·h^{-1}	[3]
Lower heating values of steam	2,676	kJ·kg^{-1}	[3]
Standard coal equivalent coefficient of coal	0.9	kg·kg^{-1}	[3]
Standard coal equivalent coefficient of oil	1.4286	kg·kg^{-1}	[3]
Standard coal equivalent coefficient of natural gas	1.33	kg·m^{-3}	[3]
Standard coal equivalent coefficient of steam	0.09	kg·kg^{-1}	[3]
Standard coal equivalent coefficient of electricity	0.1229	kg·kW·h^{-1}	[3]
EF of CO_2 in coal mining	0.00619	t·t^{-1}	[4]
EF of CO_2 in oil mining	0.0804	t·t^{-1}	[4]
EF of CO_2 in natural gas mining	0.0000748	t·m^{-3}	[4]
EF of CO_2 in diesel oil production	0.22	t·t^{-1}	[4]
EF of CH_4 in coal mining	9.32	kg·t^{-1}	[4]
EF of CH_4 in oil mining	0.00786	kg·t^{-1}	[4]
EF of CH_4 in natural gas mining	0.00000732	kg·m^{-3}	[4]
EF of CH_4 in diesel oil production	0.215	kg·t^{-1}	[4]
EF of GHG in producing electricity by coal	1,303	g CO_2-eq kW·h^{-1}	[5]
EF of GHG in producing electricity by hydro power	243.1	g CO_2-eq kW·h^{-1}	[5]
EF of GHG in producing electricity by nuclear power	13.7	g CO_2-eq kW·h^{-1}	[5]
Proportion of coal power electricity	77.18	%	[5]
Proportion of hydro power electricity	17.77	%	[5]
Proportion of nuclear power electricity	1.22	%	[5]
Energy consumption of transport by train (diesel oil power)	2.46	kg·kt·km^{-1}	[6]
Energy consumption of transport by train (electric power)	10.95	kW·h·kt·km^{-1}	[6]
Daily output of transport by train (diesel oil power)	1,104	kt·km	[6]
Daily output of transport by train (electric power)	1,328	kt·km	[6]
Energy consumption of transport by truck (diesel oil power)	52	kg·kt·km^{-1}	[6]
Average transport distance of highway(all products)	69	km	[6]
Average transport distance of railway (all products)	607	km	[6]
Average transport distance of railway (oil)	912	km	[6]
Average transport distance of railway (fertilizer)	1,365	km	[6]
N content of ammonia	82.4	%	[7]
N content of urea	46	%	[7]
N content of AN	35	%	[7]
N content of ABC	18	%	[7]
N content of ACL	25	%	[7]
CO_2 fixation in urea production	0.7	kg·kg^{-1}	[7]
CO_2 fixation in ABC production	0.65	kg·kg^{-1}	[7]
N_2O EF of nitric acid production	0.008	kg·kg^{-1}	[7]
N_2O EF of volatilization N	0.01	kg N_2O-N(kg NH_3-N + NO_x-N)$^{-1}$	[2]
N_2O EF of leaching N	0.0075	kg N_2O-N(kg leaching N)$^{-1}$	[2]

EF, emission factor; GWP, global warming potential; kt, kilotonne; TJ, terajoule.

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